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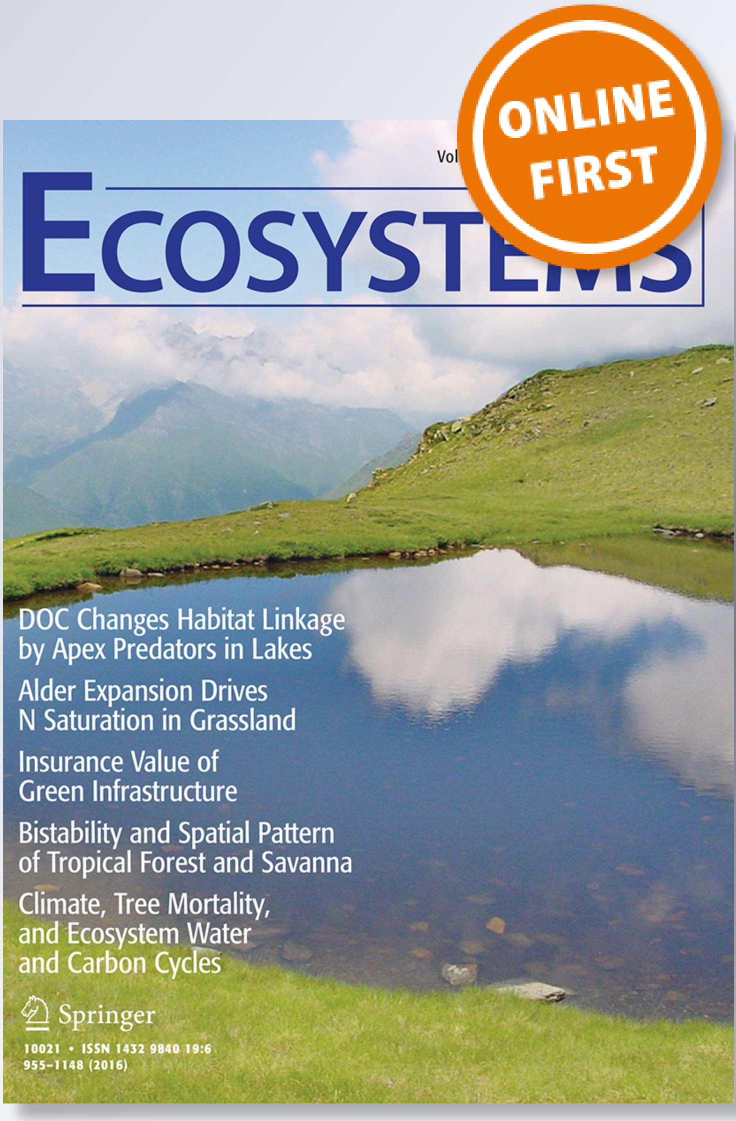

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Managed Wildfire Effects on Forest Resilience and Water in the Sierra Nevada

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Abstract

Fire suppression in many dry forest types has left a legacy of dense, homogeneous forests. Such landscapes have high water demands and fuel loads, and when burned can result in catastrophically large fires. These characteristics are undesirable in the face of projected warming and drying in the western US. Alternative forest and fire treatments based on *managed wildfire*—a regime in which fires are allowed to burn naturally and only suppressed under defined management conditions—offer a potential strategy to ameliorate the effects of fire suppression. Understanding the long-term effects of this strategy on vegetation, water, and forest resilience is increasingly important as the use of managed wildfire becomes more widely accepted. The Illilouette Creek Basin in Yosemite National Park has experienced 40 years of managed wildfire, reducing forest cover by 22%, and increasing meadow areas by 200% and shrublands by 24%. Statistical upscaling of 3300 soil moisture observations made since 2013 suggests that large increases in wetness occurred in sites where fire caused transitions from forests to dense meadows. The runoff ratio (ratio of annual runoff to precipitation) from the basin appears to be increasing or stable since 1973, compared to declines in runoff ratio for nearby, unburned watersheds. Managed wildfire appears to increase landscape heterogeneity, and likely improves resilience to disturbances, such as fire and drought, although more detailed analysis of fire effects on basin-scale hydrology is needed.

Key words: forest structure; montane; hydrology; mixed conifer; meadow; wildfire; resilience; soil moisture; fire ecology; wildland fire use.

Introduction

Projected warming and drying of the western United States climate are expected to simultaneously increase fire risks in the Sierra Nevada, and to reduce the winter snowpack that feeds 60% of California's contemporary water supply (Barnett and others 2004; Californian Department of Water Resources 2008; Westerling and Bryant 2008; Goulden and Bales 2014; Dettinger and Anderson

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Author Contributions All authors contributed to the writing of this manuscript. Gabrielle Boisramé gathered and analyzed data. Sally Thompson and Scott Stephens conceived of the study. Brandon Collins performed research and data analysis supporting the work presented here.

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2015). Fire risk and water supply in the Sierra Nevada (and much of the western United States) are linked together by the condition and function of the forests that cover approximately 70% of the montane landscapes, provide fuel to fires, and regulate hydrological processes including interception, surface energy balance, and transpiration (Soulard 2015). These forests are also vulnerable to climatic extremes. For example, ongoing drought led to the death of up to 10.5 million conifers in the southern Sierra Nevada in 2015 alone (Moore 2015), with complex consequences for future fire risk, wildlife habitat, and hydrology. Forest management has the potential to alter the relationships between forest condition, forest hydrology, and fire risk in the Sierra Nevada through manipulating tree densities, tree spatial patterns, and surface fuel loads. In particular, fire suppression, which has been practiced in the Sierra Nevada for the past century, has had negative impacts on forest resilience (the ability of a system to absorb impacts before a threshold is reached, where the system changes into a different state, Gunderson 2000), raising the question of whether alternative management strategies could improve the montane forests' ability to cope with disturbances, including those posed by climate change (Holling and Meffe 1996).

Fire suppression in the Sierra Nevada has increased the fractional forest cover of the landscape, forest canopy density, and stand homogeneity (Scholl and Taylor 2010; Stephens and others 2015). These changes elevate total fuel loads and the risk of extreme fire (Collins and Skinner 2014; Taylor and others 2014). Forest transpiration demands and interception rates also increase with stand and canopy density. Goulden and Bales (2014) showed that evapotranspiration (ET) increases exponentially with increased Normalized Difference Vegetation Index (NDVI, a measure of forest greenness and density). For example, an increase in NDVI from 0.70 to 0.77 (a 10% change) is projected to increase ET from 764 to 934 mm/y (a 22% change) (Goulden and Bales 2014). Fire suppression is also likely to result in the replacement of non-forest vegetation (particularly grasslands, shrubs, wetlands, or forbs) by trees (Lauvaux and others 2016; Norman and Taylor 2005). Such vegetation changes are likely to reduce streamflow yields (for example, Brown and others 2005; Zhang and others 2001). Indeed, contemporary Sierra Nevada mixed conifer forests transpire at rates of 760 mm/year (Bales and others 2011), as much as four times higher compared to the grasslands and meadows (Loheide and Gorelick 2005).

Assessing the net effects of vegetation change on water demand at basin scales, or at the scale of the whole Sierra Nevada, is complicated due to the variety of mechanisms through which vegetation and water interact. For example, conifers are relatively inefficient at extracting water from dry soils, and may use less water compared to drought-adapted shrub communities when water is scarce (Royce and Barbour 2001). However, coniferous forests may also obtain as much as a third of their water from fractured rock beneath the developed soil (Bales and others 2011), meaning that conclusions drawn from observations made in shallow soils could under-estimate conifer water use (Royce and Barbour 2001). The increased temperature of blackened trees and reduced shading in burned areas may also increase melting and sublimation rates of snowpack, as well as evaporation rates from bare soil, partly negating gains in water obtained from burned forests' reduced transpiration and interception (Neary and others 2005).

Overall, fire suppression facilitated an expansion and increase in the density of coniferous forests in the Sierra Nevada, which has increased fire risk, landscape homogeneity, and, probably, plant water use. In the context of a warming, drying, and increasingly fire-prone climate, these consequences are likely to reduce the resilience of the montane landscape. New forest management strategies are called for to encourage increases in forest resilience, water supply, and diversity. Forest management strategies have traditionally focused on canopy thinning (Stephens and Moghaddas 2005), prescribed burning (Fernandes and Botelho 2003), and other forms of fuel reduction (Agee and Skinner 2005). These strategies hold the primary goal of reducing surface and ladder fuel loads, with reducing plant water demand being a secondary outcome (Grant and others 2013). An alternative forest management strategy, 'managed wildfire,' has received relatively less attention (Collins and Stephens 2007). Managed wildfire uses fire as a forest treatment, but differs from conventional prescribed burning in that it relies on natural ignition events. Managers refrain from intervening with the progression of naturally occurring fires as long as there is an approved fire management plan, which specifies intervention (that is, suppression) if other management goals (safety, or air quality for instance) are threatened.

The fire characteristics associated with managed wildfire are distinct from those associated with prescribed burning. Managed wildfire encompasses more diverse and heterogeneous fire patterns, differing from prescribed burns in terms of intensity, extent, spatial pattern, severity, and burning duration (Collins and others 2011). Managed wildfires typically create high-severity burn areas in forests that can cause stand replacement which, unlike lower intensity prescribed burning, can change landscape-scale vegetation composition and structure (Collins and Stephens 2010), and increase the heterogeneity of vegetation cover (Hessburg and others 2005, 2015). Prescribed burns, thinning, and other fuel treatment options have traditionally not achieved similar increases in landscape diversity. Thanks to this heterogeneity, forests with restored fire regimes may represent a more resilient state for montane ecosystems compared to the contemporary fire-suppressed condition—at least in the sense that more resilient forests are less likely to be catastrophically altered by severe disturbance (Millar and others 2007; Kane and others 2014; Holling and Meffe 1996).

Despite these intriguing qualities, managed wildfire has been only minimally used for forest management: for instance, only two locations in the Sierra Nevada have an extensive history of managed wildfire (Collins and Stephens 2007). Obvious constraints and challenges associated with managed wildfire include identifying safe and appropriate locations for its use, transitioning to a natural fire regime from fire-suppressed conditions, and minimizing risks to people and property. In spite of these constraints, future use of managed wildfire is likely to increase. In 2015, three national forests in the southern Sierra Nevada (Sierra, Sequoia, and Inyo National Forests) have proposed over 50% of their land base for use of managed wildfire to restore more natural vegetation structure and patterns. There is relatively little information on how, and over what timescales, the introduction of such managed wildfire treatments could alter long-term landscape composition and ecosystem functioning, including water use.

We expect that managed wildfire will add heterogeneity to vegetation cover (Hessburg and others 2005, 2015). The associated reduction in fuels and competition among trees should increase forest resilience to fire and drought (Stephens and others 2009; van Mantgem and others 2016). Due to reduced interception and transpiration, we also predict increased stream flow and summer soil moisture (Brown and others 2005). Yosemite's Illilouette Creek Basin (ICB) provides a unique opportunity to evaluate these hypotheses, being, to the best of our knowledge, the only long-term managed wildfire area in the western United States coupled with a long-term stream gauge. Previous research in this watershed has focused on fire patterns (for example, Collins and Stephens 2010), forest structure and fuels (for example, Collins and others 2016; Kane and others 2014), and diversity in plant and pollinator species (Ponisio and others 2016). By combining findings from earlier studies with new datasets and analyses, this paper provides the first investigation of watershed-scale change in vegetation cover, and its effects on water resources and drought resilience within this basin. Specifically, we summarize 'lessons learned' from the 40-year application of managed wildfire in ICB, focusing on three major topics: (a) vegetation changes induced by the managed wildfire regime, (b) the effect of vegetation change on water resources in ICB, and (c) the effects of vegetation change on fire characteristics and forest resilience.

Study Area

The Illilouette Creek Basin (ICB) is a 150 km^2 basin within the Upper Merced Watershed in Yosemite National Park, California, USA (Figure 1A). It spans an 1800-m to approximately 3000-m elevation range in the central Sierra Nevada mountain range. This area experiences a Mediterranean climate. Based on observations from Remote Automated Weather Stations near ICB, average January daily minimum temperatures ranged from −5 to 1°C, whereas average July daily maximum temperatures ranged from 24 to 25°C (2000–2015; http://www.wrcc.dri.edu/; stations: White Wolf, Crane Flat). Average annual precipitation (Oct–Sep) ranged from 47 to 60 cm at these stations (for years 2000–2015), and is dominated by winter snow. The basin is covered by coniferous forests (dominated by *Pinus jeffreyi*, *Abies magnifica*, *Abies concolor*, and *Pinus contorta*), rocks, meadows, and shrublands (dominated by *Ceanothus cordulatus)* (Collins and others, 2007). This area has never experienced timber harvesting and likely had minimal impacts from livestock grazing (Collins and Stephens 2007).

Fire suppression began in ICB in the late 19th century (Collins and Stephens 2007) and continued until 1972, when Yosemite National Park began its ''Natural Fire Management'' program (van Wagtendonk 2007). The duration of the managed wildfire program at ICB and the large total proportion of the basin area that has burned since 1972 (52% of the total area and ≈75% of the vegetated area) make it an ideal place to study the landscape effects of managed wildfire. Although ICB was impacted by approximately 100 years of fire suppression (only 8 ha are known to have burned between 1880 and 1973), fire frequency and extent since the onset of the managed wildfire program in 1972 are similar to

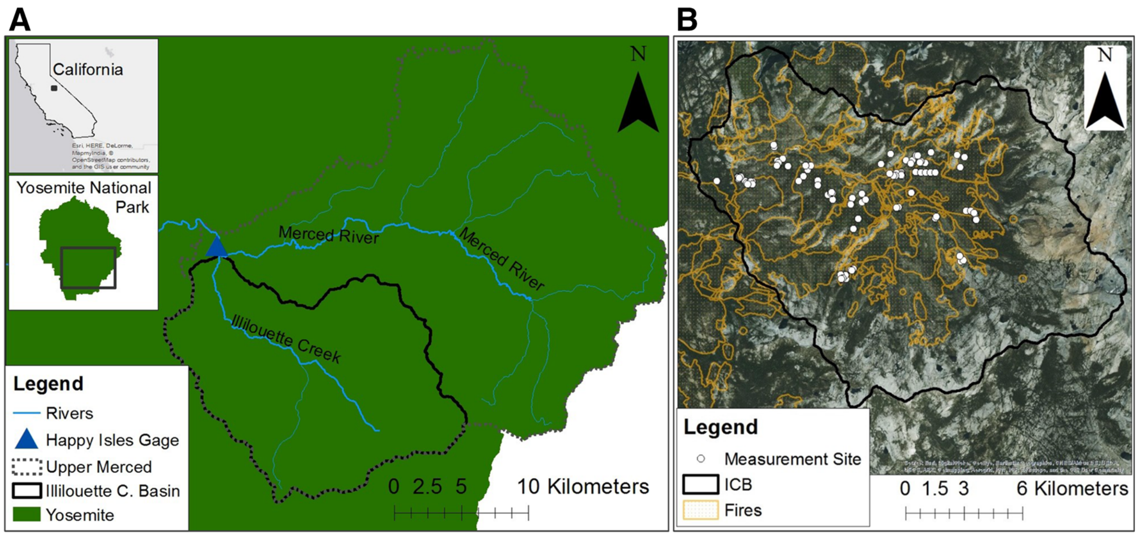

Figure 1. **A** Study area and location within Yosemite National Park and the state of California. **B** Map of the ICB with all fire boundaries since 1970, and the locations of all soil moisture measurements.

that in the non-fire-suppressed historical period (a 6.8-year recurrence interval, versus 6.2 historically, based on fire scar measurements, Collins and Stephens 2007). Figure 1B shows the extent of all fires in ICB over the past century.

Methods

Vegetation Change

Vegetation change from the fire-suppressed condition (1969) to the present managed wildfire condition (2012) was assessed using an object-oriented image analysis (eCognition) of aerial photography. The imagery sources are contemporary aerial photographs (National Agricultural Imaging Program 2012, four-band, 1-m resolution, USDA Farm Service Agency 2015) and black and white aerial photographs from 1969, capturing the basin condition near the end of the fire-suppressed period (Yosemite National Park Archive, 8-bit 0.5-m resolution, produced by Cartwright Aerial Surveys). The black and white photography was orthorectified in the ERDAS Imagine Leica Photogrammetry Suite, and then spatially and radiometrically degraded (from 8-bit, 0.5-meter resolution to 4-bit, 2-meter resolution) to improve the performance and processing speed of the classification algorithms (Caridade and others 2008). When two or more images overlapped, the best of those images was selected manually for classification based on the clarity of individual objects.

The orthorectified photos were classified as granite, water, mixed conifer forest, shrub, sparse meadows, aspen, and dense meadows. Meadows are defined as areas dominated by grasses and forbs; dense meadows have little to no bare ground and appear green in summer aerial photographs; while sparse meadows have larger amounts of bare ground and appear brown. The dense meadow category encompasses wetlands, but the aerial image analysis was not able to reliably separate true wetlands from areas with dense summer green grass. Granite was identified using the 2012 imagery first, where it is more easily distinguishable from grassland compared to the black and white imagery. Mapped granite outcrops from 2012 were then applied to the 1969 map, under the rationale that fire would not affect the distribution of rocky land cover. The 2012 classification was validated using 274 ground-truth points mapped in 2013–2015. The 1969 vegetation class maps were validated by randomly selecting point locations and comparing a visual classification of the points to the automated classification. Greater than 90% accuracy in the classification was achieved. Changes in landscape composition between the fire-suppressed and the contemporary conditions were assessed in terms of landscape composition, patch size, and Shannon's evenness index, using FRAGSTAT (McGarigal and others 2012). Total cover was calculated for each vegetation type in each year, accounting for slope of the landscape. Classification uncertainty was propagated into the change estimates (following Congalton and Green 2008).

Forest Mortality During Drought

The US Forest Service maps new tree mortality (defined by yellow to reddish brown trees) in the

Sierra Nevada every summer using aerial surveys (Moore 2015). These mapping surveys have high levels of accuracy: Only 4% of tree mortality or injury was missed, and damage type was identified correctly 83% of the time (Coleman and others 2015).

We used these data to compare drought-associated tree mortality (not caused by fire) between ICB and nearby forests in 2014 and 2015. The year 2014 was when the Forest Service first observed a large increase in tree mortality related to the current drought (Moore 2015). Although most mortality was attributed to insects or diseases, such as mountain pine beetle (*Dendroctonus ponderosae*) and cytospora canker(*Cytospora kunzei*), the susceptibility of the trees to mortality from these stressors was likely increased by drought (Allen and others 2010). Although a full analysis of the effects of fire history on drought-related forest mortality would require explicitly accounting for dispersal rates of beetles and other damaging agents, detailed localized weather, and groundwater availability, such an analysis is beyond the scope of this paper. Instead, we compare mortality within ICB to multiple control watersheds: (a) the control watersheds described below under ''Runoff ratio analysis,'' and (b) comparable watersheds adjacent to ICB having had less than 15% of their area burn since 1994. All control areas lie within the same elevation and climatic zones as ICB, and thus should experience similar drought stress, and they have all experienced fire over less than 15% of their area in the past 20 years. The watersheds used here are defined using the 12-unit watershed delineations available from USGS (http://nhd.usgs.gov/wbd.html), and are all approximately the same size (48–193 km^2, compared to 158 km^2 for the ICB). For each of these control areas, we calculated the drought-related mortality per km^2 of forest by dividing the number of dead trees by the total forested area (defined using the LANDFIRE existing vegetation type layer, LANDFIRE, 2012b).

Soil Moisture Measurements

The changes in vegetation structure induced by the managed wildfire regime were expected to lead to changes in the local water balance. A location's vegetation cover frequently indicates local hydrological conditions favoring that vegetation type (for example, Mountford and Chapman 1993; Araya and others 2011; Milledge and others 2013), and different plant water use profiles, rooting depths, and canopy structures also alter micrometeorological conditions beneath the canopy and change the local water balance (Zhang and others 2001; Brown and others 2005; Rambo and North 2009; Ma and others 2010). Therefore, vegetation and soil water storage are expected to co-vary.

Surface soil moisture was comprehensively mapped in ICB in the summers of 2014–2015. Over 3300 measurements were made in more than 70 sites, selected to cover representative conditions, including: burn severity, time since fire, soil type, slope, aspect, elevation, and vegetation cover (Figure 1B). The ranges of values for physical variables within the watershed and measurement locations are compared in the supplementary material, Table S.1.

Each site was measured between one and five separate times over the 2-year study, capturing both early and late summer moisture when possible. Measurements were made with a 12-cm time-domain reflectometer (TDR) Hydrosense II probe (Campbell Scientific 2015). All measurements were recorded in terms of volumetric water content (VWC). The VWC is the proportion of the total volume of the soil matrix that consists of water, ranging from 0 for completely dry soils to approximately 0.6 for saturated, highly porous soils (pure water would have a VWC of 1). Soil moisture was related to site condition using a random forest model (Liaw and Wiener 2015), which was subsequently used to upscale the soil moisture results to the whole basin, using geospatial data and the vegetation maps. For modeling purposes, vegetation cover was assigned based on the broad vegetation classes inferred from the aerial imagery: sparse meadow, dense meadow, mature conifer, conifer recruitment, and shrub.

The random forest model predicts soil moisture (as a continuous value) using the following predictor variables: vegetation type, upslope area, slope, aspect, topographic position index (Weiss 2001), topographic wetness index, distance from nearest stream, years since last fire, and maximum fire severity. To model change in soil moisture as a consequence of fire, we ran the model under 1969 (fire suppressed) and 2012 conditions. The first model run used the 1969 vegetation map, time since fire set to 100 (reflecting the duration of fire suppression), and both times burned and fire severity set to 0. The second model version used the 2012 vegetation map and actual fire data as of 2012. We then subtracted the modeled VWC values for 1969 from the modeled 2012 values to calculate the change. Note that this model does not include meteorological data, and thus the model results represent the change we would expect under identical climatic conditions with only the fire history and vegetation cover being different.

Random forest models predict a continuous variable by creating a large number of regression trees, using a random subset of all possible predictor variables to create each tree, and then taking the average predicted value of all of the trees. Each regression tree divides the data into smaller and smaller groups, or nodes, until a stopping criterion is reached. At each step, the data in one node are divided into two more nodes based on the division in the predictors that creates the largest separation in values between the two groups. The value of a new point in the variable space is computed by following the path from the first node to the appropriate terminal node. This method avoids issues of overfitting that can result from using only one regression tree, and allows for fitting non-linear responses between variables and predictors (Grömping 2009; Kane and others 2015). We used the randomForest package in the R program to fit the model, and set the minimum node size to 5 and the number of trees to 100 (100 trees minimized the RMSE of the model) (Liaw and Wiener 2015). All data were randomly divided into a training dataset (75% of all data) and a test dataset (25%).

The ''years since fire'' variable was calculated using a digital fire atlas consisting of all fire perimeters for which a record exists, dating back to 1930 within Yosemite National Park (available from https://irma.nps.gov/Portal). The fire atlases are a best approximation of actual burn perimeters, but do not provide information on the spatial heterogeneity of burning within fire areas (Morgan and others 2001). We used satellite-based estimates of fire severity to characterize this heterogeneity within fire areas for all fires over 100 ha since 1972. For fires that occurred in 1984 or later (n = 11), we used a relative version of the differenced Normalized Burn Ratio (RdNBR), derived from Landsat Thematic Mapper images (Miller and Thode 2007). Fire severity data for fires prior to 1984 (n = 8) were derived from Landsat MSS images, which only had 4 bands (as compared to 7 bands on the Landsat Thematic Mapper sensor), and had larger pixels (66 m, as compared to 30 m for the later sensor). As a result, fire severity data from fires that occurred between 1972 and 1983 were developed using a relative version of the difference between pre- and post-fire Normalized Difference Vegetation Index (RdNDVI) (Collins and others 2009; Thode 2005). Fire severity class (unchanged, low, moderate, and high) for each soil moisture measurement site was extracted from the nearest 30-m (or 66-m) resolution raster cell. Thresholds for RdNBR fire severity classes were taken from Miller and Thode (2007). Thresholds for RdNDVI were developed from the RdNBR thresholds because we did not have historical field data to calibrate MSS images. In pre-1984 fire locations that had reburned after 1984, we used post-1984 fire severity because it is more accurate and likely to be more relevant to soil moisture conditions measured in 2014–2015.

Runoff ratio analysis

Runoff from the ICB is first gauged at the Happy Isles Gauge on the Upper Merced Watershed, where the ICB comprises approximately 30% of the gauged area. To determine if any changes in flow characteristics could be discerned at Happy Isles and potentially be attributed to changes in ICB, we divided the flow time series into pre- and post-1973 time periods, and normalized annual (water year, October–September) flow by annual precipitation to obtain the annual runoff ratio. Trends and changes in the distribution of annual runoff ratio were computed for both time periods at Happy Isles, and for three nearby USGS stream gauges measuring flow from control watersheds with comparable area, vegetation, topography, and elevation, but where fire suppression has continued. The gauges are Upper Merced River at Happy

Table 1. All Watersheds are of Comparable Size, Elevation Range, Median Streamflow, Annual Precipitation (PPT), and Percent Vegetated Area (from LANDFIRE)

Watershed name	Area (km^2)	Elevation (m)	Flow (m^3/s)	PPT (m)	Veg (%)	Burned (%)	Earliest Data
Upper Merced	453	1200–3700	2.9	1.2	76	23.0	1915
MF Stanislaus	119	1900–3400	1.8	1.5	55	0.2	1938
SF Stanislaus	112	1600–2900	1.7	1.6	88	3.3	1913
Cole Creek	53	1000–2600	0.4	1.5	91	14.7*	1927

*The ''Burned (%)'' column gives the percent of the area of each watershed known to have burned since 1930. The control watersheds have all had very little area burned in recorded history, and have only experienced one or two fires compared to 74 in the Upper Merced Watershed. Streamflow data are available for at least 35 years before the ICB's change in fire management (as shown in the ''Earliest Data'' column which gives the first year of streamflow data). Annual precipitation for this table is calculated from PRISM. *The 14.7% burned area in Cole Creek is due to a fire in 2004; prior to this only 1% of the watershed had been burned.*

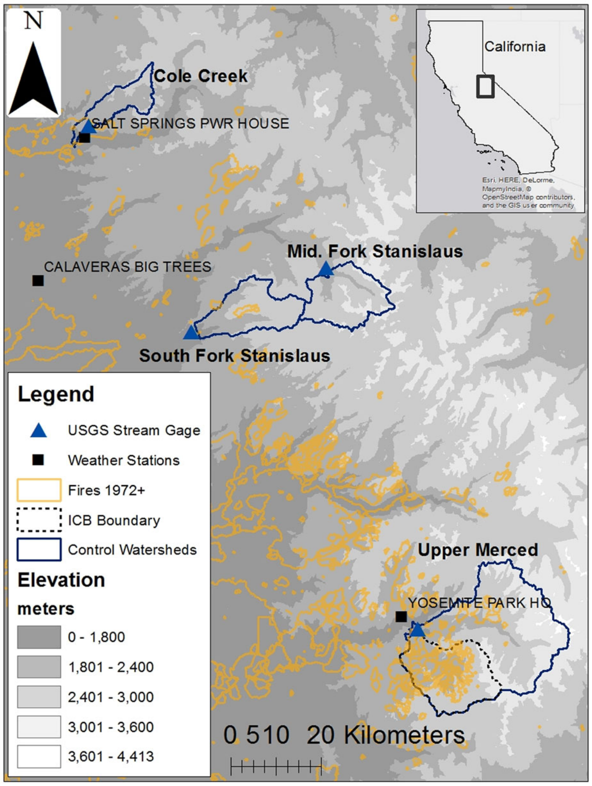

Figure 2. Map of control watersheds, stream gauge locations, and the locations of the closest weather stations with records of over 10 years within each time period (pre- and post-1974) shown in the context of elevation and fires occurring after 1972.

Isles (USGS #11264500), Middle Fork Stanislaus River at Kennedy Meadows (#11292000), South Fork Stanislaus River at Strawberry (#11296500), and Cole Creek near Salt Springs Dam (#11315000). Approximately 25% of flow at Happy Isles is from Illilouette Creek (James Roche personal communication). Details for all watersheds are given in Table 1, showing that the control watersheds have comparable elevation ranges, climate, and vegetation cover to the Upper Merced watershed. Locations of the control watersheds, stream gauges, and weather stations are shown in Figure 2. We also compared the ecological similarity of the watersheds using the LANDFIRE Biophysical Settings layer, which models potential vegetation under historic fire regimes prior to European settlement, taking into account climate, substrate, and topography (Rollins 2009). The most common potential vegetation category for all watersheds is Mediterranean California Red Fir Forest, suggesting that the watersheds are comparable from a biophysical perspective (LANDFIRE 2012a). Precipitation is sparsely gauged in the Sierra Nevada, so we conducted the analysis with 3 precipitation datasets: the point-scale measurements made at the weather station closest to each basin, and gridded climate products PRISM (Oregon State University 2004) and ClimSurf (Alvarez and others 2014).

Results

Vegetation Change Following the Introduction of Managed Wildfire

Following the introduction of managed wildfire, the vegetation cover in ICB became more heterogeneous and less dominated by coniferous forest (Figure 3). Figure 3 Panels C and D show examples of common transitions that occurred during the managed wildfire regime: coniferous forest to shrubland (C1–C2), and coniferous forest to a mix of dense meadows and sparse grassland (D1-D2). Quantitatively, forest cover declined by 22% (from 109 to 84 km^2), whereas the area of meadow vegetation increased by 200% (0.8–2.4 km^2) and shrublands by 24% (14–17 km^2) (Figure 4).

The organization of the landscape also changed following the introduction of the managed wildfire regime. Today, the ICB contains a greater number of distinct vegetation patches, with a wider range of size characteristics compared to those in 1969. Forest patches declined in area (for example, the largest forest patch decreased by 38%, from 51 to 32% of the basin area), and other vegetation type patches expanded (for example, the largest shrub patch increased from 0.4 to 1.7% of the basin area, and the largest meadow patch increased from 0.1 to 0.9% of the basin area). The basin-scale Shannon's evenness index, which summarizes the structural diversity in the landscape on a scale from 0 to 1, increased from 0.65 to 0.80 from 1969 to 2012, consistent with an increase in structural diversity.

Effects on soil moisture and hydrology

In ICB, vegetation type was closely associated with the summer surface soil moisture content (Figure 5). Dense meadows had the highest median VWC throughout the summer (0.33 in May to 0.14 in August). At the beginning of the summer, conifer patches had the next highest median VWC (0.15), but were comparable to other dry vegetation by the end of summer, at 0.04 VWC. Shrublands and sparse meadows had comparable soil moisture values at the beginning of summer (0.11 and 0.12, respectively), and end of summer (with 0.03 and 0.05, respectively).

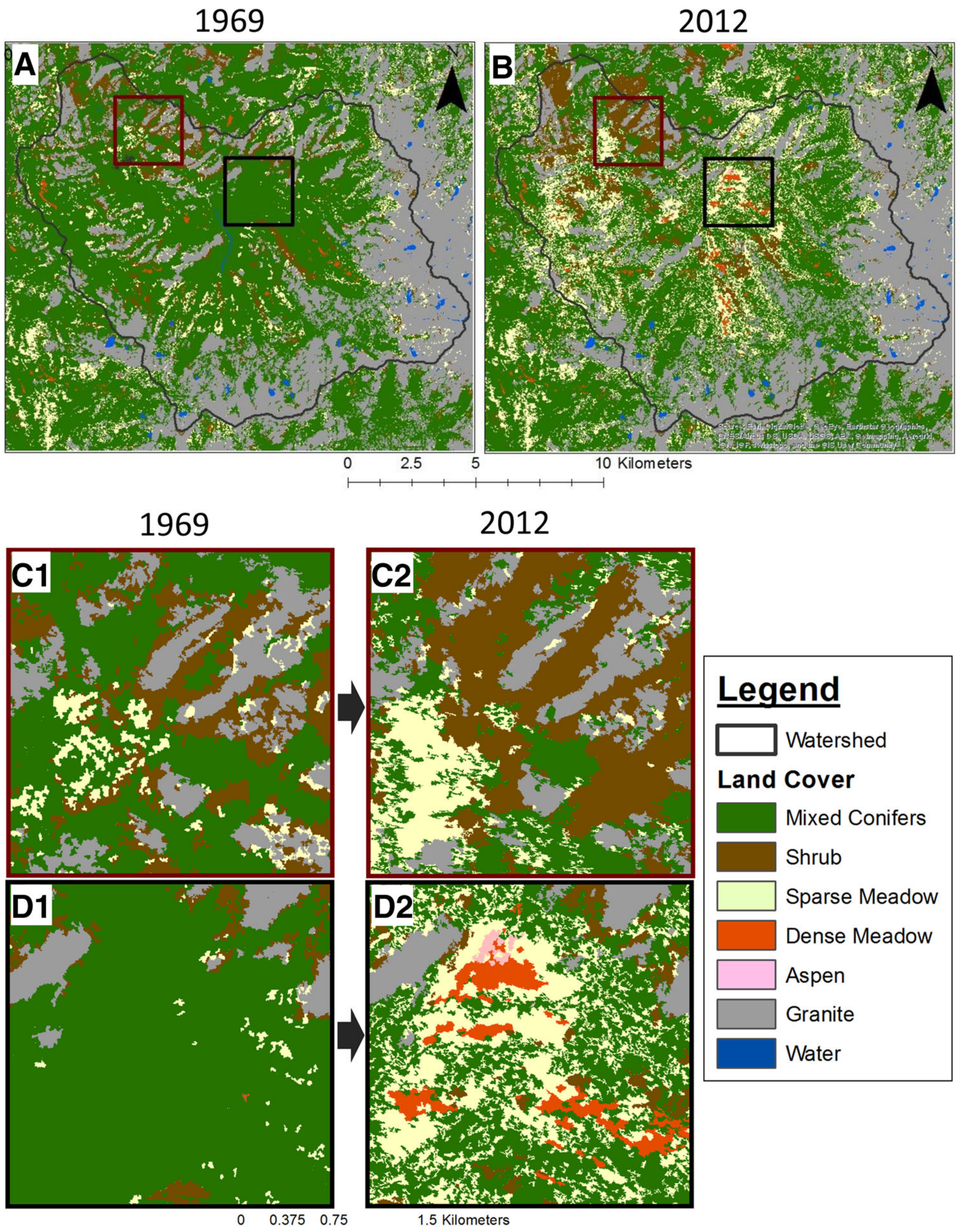

Figure 3. Maps of land cover in the Illilouette Creek Basin in 1969 (after 100 years of fire suppression) and 2012 (40 years after fire regime change). *Insets* show an area where both sparse grasslands and shrublands have expanded post-fire (C1, C2), and another that has generally changed from conifer cover to more open vegetation (D1, D2).

We used a random forest model to isolate the effect of vegetation on local summer soil moisture. The model results predicted the observed soil moisture values with an RMSE of 0.07 and coefficient of determination (r^2) of 0.76, with vegetation significantly predicting differences between sites. We found that the random forest model performed best when it was broken up into two separate sub-models: one for early summer and one for late summer. We trained the early summer sub-model on data taken in late May and early June, and the late summer sub-model using data from late July and early August.

We used the model to explore potential basin-scale effects of the mapped vegetation changes on soil moisture in ICB. At the whole basin scale, the model predicted minimal changes in spatially averaged soil moisture, despite the large changes in vegetation; these changes were slightly positive for June (ΔVWC = 0.003, an 8% change), and negative for July (ΔVWC = −0.004, a 4% change). Dramatic changes, however, were predicted in the

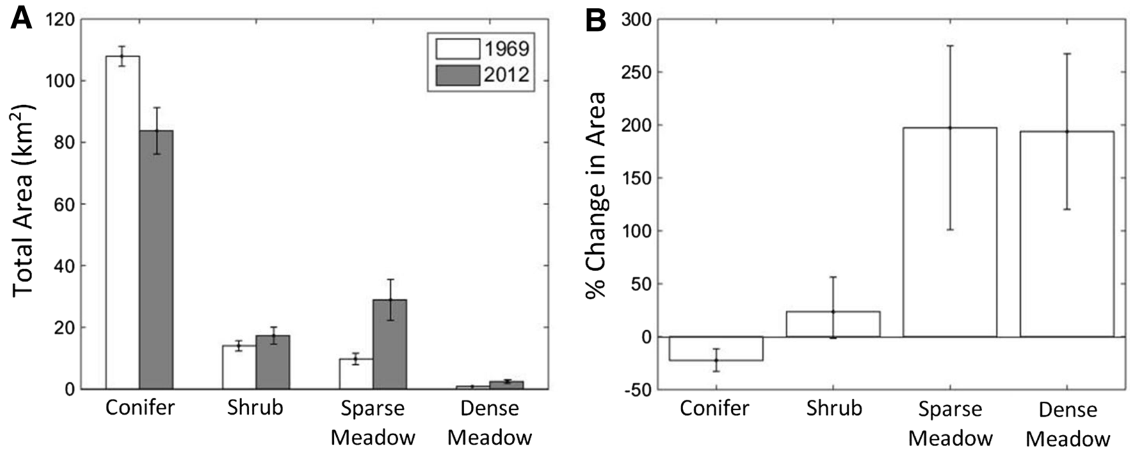

Figure 4. Change in vegetation cover in terms of total area covered in each year (**A**) and as a percent change in area covered (**B**). *Error bars* represent the level of uncertainty in the vegetation mapping. For example, if conifers were mapped with 90% accuracy, then the *error bars* for conifer would show ±10% of the area calculated. Aspen is not included due to difficulty identifying aspen in the black and white 1969 images making quantification of change in area highly uncertain.

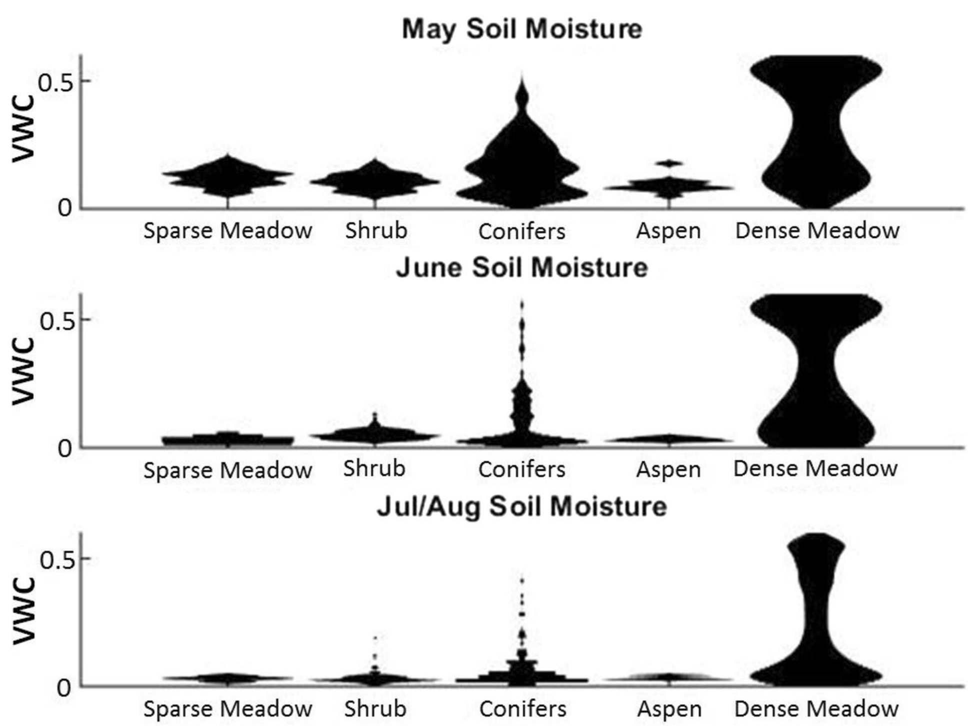

Figure 5. Plots of the distribution of soil moisture under each vegetation class for the same measurement locations in May, June, and July/August for 2014 and 2015. The results are divided between locations dominated by sparsely vegetated meadows, shrubs, conifers, aspen, and dense meadows. Soil moisture is given as the volumetric water content (VWC).

wetness of individual sites. Figure 6 shows the change in mean summer VWC (calculated by averaging early and late summer values) between the 1969 vegetation conditions and the 2012 vegetation plus fire history. The site-specific effects of slope, aspect, upslope area, and elevation were explicitly controlled for. Burned forest sites that regenerated with dense meadow (and potentially wetland) vegetation increased in predicted summer VWC by as much as 0.3 (for comparison, some of the most highly saturated sites had a VWC of approximately 0.5). These large local increases in

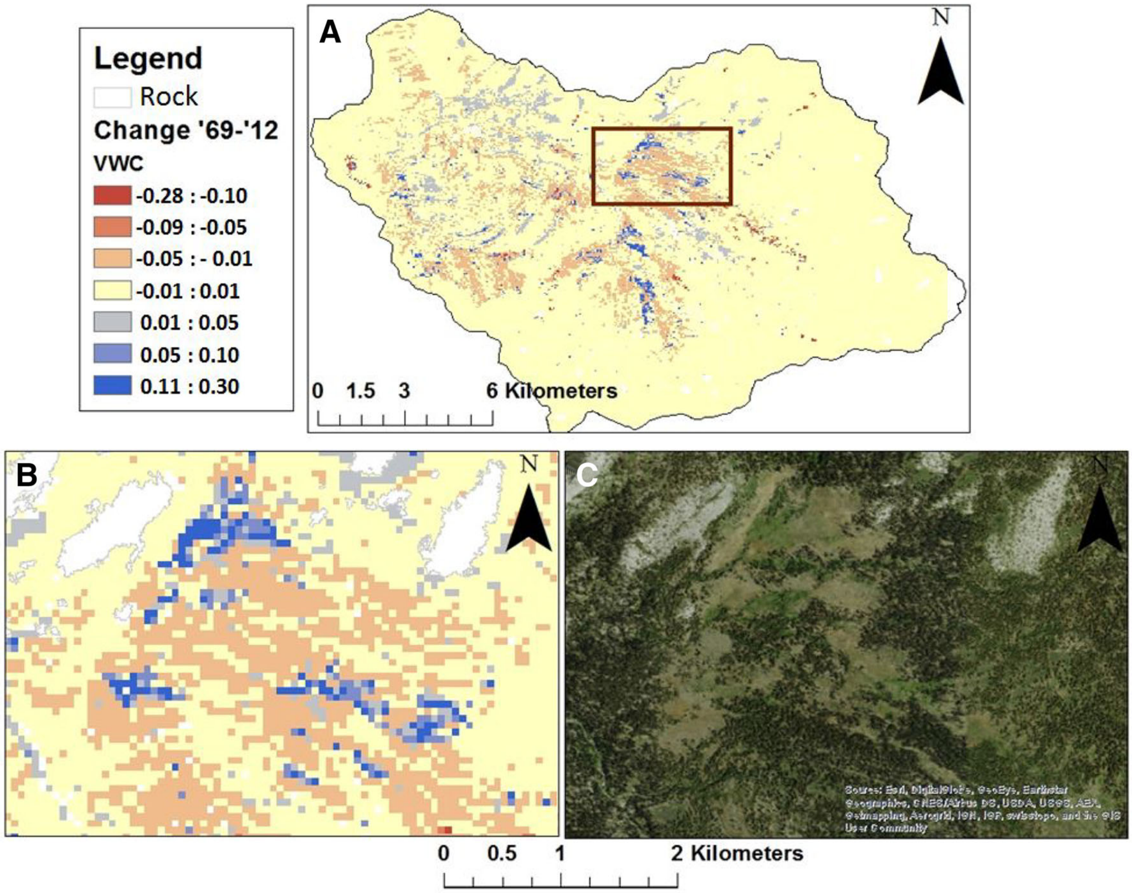

Figure 6. **A** Map of ICB showing change in summer mean volumetric water content for the top 12 cm of soil from 1969 to 2012, calculated using a random forest model. This probabilistic model does not include any information on weather, only vegetation, topography, and fire history. A positive VWC change (*blue*) indicates an increase in water storage, while a negative VWC change (*red*) indicates a decrease. **B** Close-up of VWC change in an area with a variety of changes in soil moisture. **C** 2014 aerial imagery (Esri inc.) of the region shown in (**B**) with extent given in (**A**) (Color figure online).

Table 2. Percent Change in Median Pre-1973 and Post-1973 Annual Runoff Ratio (Total Streamflow Divided by Total Precipitation) Using Three Sources of Precipitation Data: Remote Weather Stations and Gridded Precipitation Estimates from PRISM and ClimSurf

Watershed	Weather Stations (1940–2000)		PRISM (1940–2012)		ClimSurf (1950–2000)	
	Change (%)	*p* value	Change (%)	*p* value	Change (%)	*p* value
Upper Merced	13	0.43	2	0.36	0.0	0.65
MF Stanislaus	−6	0.26	−1	0.60	−6	0.05
SF Stanislaus	−8	0.41	−4	0.66	−9	0.06
Cole Creek	−7	0.00	−11	0.05	−12	0.02

Depending on the data source, start dates for the pre-1973 era are either 1940 or 1950, and end dates for the post-1973 era are either 2000 or 2012. Using only data from 1950-2000 for all datasets does not alter the sign of the change in runoff ratio for any of the watersheds, and only slightly changes the magnitude and p-value. All watersheds show a decrease in runoff ratio after 1973 except for the Upper Merced Watershed (which includes ICB).

Table 3. Summary of Drought-related Tree Mortality During the Summers of 2014 and 2015 in the ICB Compared to Similar Watersheds with Fire-Suppressed Landscapes

Region	Conifer area (km^2)	Dead Trees, 2014	Dead/km^2, 2014	Dead Trees, 2015	Dead/km^2, 2015	Ratio, 2014	Ratio, 2015
ICB	80.6	325	4.0	1040	12.9	1.0	1.0
Table 1 Watersheds							
MF Stanislaus	39.3	351	8.9	540	13.8	2.2	1.1
SF Stanislaus	71.7	11024	153.8	23352	325.8	38.1	25.2
Cole Creek	36.0	544	15.1	6294	174.6	3.7	13.5
Adjacent Watersheds in same elevation range							
Bridalveil	49.5	6094	123.0	33505	676.9	30.5	52.4
Chilnualna	34.7	306	8.8	10579	304.8	2.2	23.6
USF Merced	87.6	1787	20.4	5348	61.0	5.1	4.7
Total	171.8	8187	47.6	49432	287.7	11.8	22.3

These watersheds include both the control watersheds used in the runoff ratio analysis as well as fire-suppressed watersheds adjacent to ICB and falling within the same elevation range. The ''Ratio'' columns give the proportion of tree mortality density in a given region to the density in the ICB, showing that all regions have a higher density of drought-related tree mortality compared to the ICB.

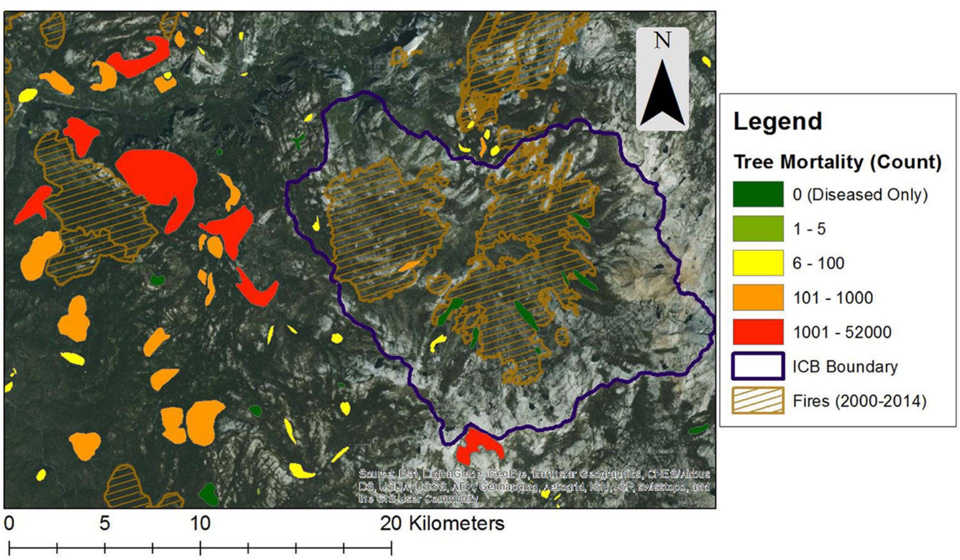

Figure 7. Map of drought-related tree mortality and disease areas in 2014, along with all fires since year 2000. The number of recently dead trees is much higher outside the ICB than within it. There is very little overlap between burned sites and large mortality patches, despite burned areas still containing many large trees (as shown in this aerial image from 2014, provided by Esri inc.).

soil moisture were offset by widespread, minor decreases in soil moisture; the greatest decreases were modeled in locations where conifers encroached on meadows.

The pre- and post-1973 runoff ratio changes computed using three different precipitation datasets for the ICB and three control (unburned) watersheds are shown in Table 2. The sign of any detected change in runoff ratio differs between the Upper Merced and the control basins, with Upper Merced tending towards relatively stable or positive trends in runoff ratio, and the other basins either decreasing or stable. The significance of observed trends is not consistent between basins and datasets.

Effects on Tree Mortality During Drought

Aerial surveys by the USFS during the severe drought years of 2014–2015 show minimal levels of tree mortality within ICB, despite extensive tree death due to beetles and other drought-associated causes of mortality in adjacent, mainly unburned areas. These surveys match our qualitative field observations of tree health in the area. In 2014, approximately 47 trees per km^2 of forest died within fire-suppressed watersheds adjacent to ICB that were climatically similar, compared to only 4 dead trees per km^2 of forest within ICB. Table 3 gives a summary of tree mortality in similar areas, showing that ICB has the lowest mortality in this set of watersheds. Suggestively, areas of drought-related tree mortality are mostly located in unburned regions. Figure 7 shows an example of an area adjacent to ICB with 2014 beetle-killed tree areas occurring at the edges of previously burned areas, but not overlapping with them. The only mapped tree mortality within the burned portion of the ICB occurred in areas that had experienced only low severity fires, and the largest patch was only 20 ha in extent, compared to patches of mortality up to 680 ha located within 10 km of the ICB. (For more details, see supplementary material Figure S.1.)

Discussion

Managed wildfire in ICB has dramatically altered the composition and structure of the landscape. At the end of the fire-suppressed period, the vegetated parts of the basin formed essentially a single, homogeneous forested patch (Figures 3A, 4A). Fire has fragmented the vegetation cover in ICB, resulting in an increase in the distribution of vegetation patch sizes, and an increase in the proportion of the basin covered by meadow and shrub vegetation types. Changes in understory vegetation and in forest structure were not captured by our analysis of the historical imagery, though they have been observed in other studies within the ICB (Collins and others 2016; Ponisio and others 2016; Kane and others 2015). The bulk landscape classification therefore presents a conservative estimate of the true degree of vegetation change in the ICB.

The change in vegetation composition and organization in the ICB also changed the patterns of fire occurrence and effects. Contemporary fires in the ICB occur with relatively high frequency, but are predominantly of low to moderate severity, interspersed with relatively small patches of stand-replacing fire (Collins and Stephens 2010). This pattern is consistent with our understanding of how historical fires burned in these landscapes (Collins and Stephens 2007; Collins and others 2009; van Wagtendonk and others 2012; Stephens and others 2015). Similar fire patterns are not observed in comparable forests where fire suppression continues (Miller and others 2012). Fires in suppressed forests create uncharacteristically large stand-replacing patches, and burn with a greater proportion of stand-replacing effects (Miller and others 2009; Miller and Safford 2012), consistent with the theoretical prediction that removing natural variability from a landscape reduces its ability to recover when disturbance (for example, catastrophic fire, drought, or insect outbreak) eventually occurs (Holling and Meffe 1996; Holling 2001).

Two main factors drive the divergent fire patterns in the ICB relative to the fire-suppressed Sierra Nevada. First, in forests with continued fire suppression, the only fires that burn a significant area are those that ''escape'' initial suppression efforts. ''Escape'' fires tend to burn under extreme weather conditions, compounding the already fuel-loaded condition of many Sierra Nevada forests, leading to uncharacteristically extensive and severe wildfires (Finney and others 2011; North and others 2015). In contrast, the fires in ICB burn under a broad range of fire weather conditions (Collins and others 2007; Miller and others 2012). Second, the frequency and extent of fires in the ICB is such that fuel consumption by previous fires limits the spread and intensity of subsequent wildfires (Collins and others 2009; van Wagtendonk and others 2012; Parks and others 2014). This 'self-limiting' characteristic of the fires is directly linked to the increased heterogeneity and more even patch-size distribution of vegetation in ICB (Figure 3)—a more heterogeneous landscape results in more obstacles to fire spread, while reductions in forest cover reduce the landscape-scale fuel loads.

Intriguingly, the ICB may also exhibit resilience to *other* forms of disturbance. Although the 2011–2015 drought conditions in California are estimated to have killed over 10 million trees in the southern Sierra Nevada (Moore 2015), forest mortality in ICB during this period appears to be minimal. The only control watershed with comparably low drought mortality was the Middle Fork Stanislaus, which was likely less susceptible to drought because it is in the higher end of the elevation range. Other watersheds had up to 52 times higher rates of drought mortality compared to the ICB. This low incidence of drought mortality in burned areas is consistent with van Mantgem and others (2016), who found that, during the drought year of 2014, burned stands under 2100 m in elevation had a lower occurrence of tree mortality compared to areas that had not experienced fire in over 100 years.

In addition, we observed multiple persistent wetlands throughout the record-breaking drought summers of 2014 and 2015. Although it would be premature to attribute the low drought-related forest mortality rates or wetland persistence in ICB directly to the managed wildfire regime, such drought resilience is consistent with expected effects of reducing forest extent, forest density, and understory vegetation, all of which would reduce competition for limited water supplies (Grant and others 2013).

Observations to date suggest that vegetation cover is meaningfully associated with summer soil moisture in ICB. Introduction of the managed wildfire regime led to the expansion of densely vegetated meadow areas in which surface water availability is likely much higher compared to the previously forested state. Indeed, we regularly found instances of wetland vegetation regenerating amidst burned conifer stems; and yet the coniferous forest sites we measured never exhibited the summer-long saturated soil conditions found in wet meadows. We hypothesize that fire suppression enabled woody plant invasion and desiccation of meadow margins (due to increased plant water uptake), and that the reintroduction of fire has reopened those meadow regions, providing new habitat for wetland plants and year-round water sources (Norman and Taylor 2005).

The basin-scale consequences of the observed changes in vegetation remain unclear, despite data that tentatively suggest the ICB is maintaining or increasing water yields, while similar but unburned basins are reducing water yields. Further investigations of the effects of vegetation change on water balance are clearly essential. Relevant issues include understanding how snowpack and soil moisture dynamics respond to the different vegetation types, and how patch-scale changes in vegetation and water availability scale up to produce changes in runoff response.

Although the measured differences in runoff ratio changes between the Upper Merced River and the control watersheds are small, they represent a lower bound on the influence of wildfires on this area's stream flow: Any slight flow regime change at Happy Isles related to wildfire is expected to stem from a larger change (proportionally) within ICB, because ICB is the sub-watershed with the greatest proportion of burned land feeding into the Happy Isles Gauge. Tentatively, the managed wildfire regime may have either stabilized or increased the runoff yield from ICB, but further analysis is needed to test this hypothesis. Ongoing work focuses on basin-scale hydrologic modeling that propagates the changes in vegetation structure in ICB into estimated hydrologic response, and allows us to explore how these changes might propagate into overall watershed function.

Our paired watershed analysis is promising, but contains a high level of uncertainty. Unfortunately, all three control watersheds are smaller compared to ICB and located further north. We looked into using several larger and more southern watersheds, but they all either had large gaps in the flow record or were strongly affected by reservoirs (such as the Tuolumne River below Hetch Hetchy). We believe that runoff ratios should be comparable between these watersheds, however, because weather variations over time are similar among all watersheds being compared. The watersheds also have similar elevation ranges and are all on the western side of the central Sierra Nevada, and thus should experience similar proportions of snow versus rain as well as similar evaporative demand. These watersheds also all have streamflow records spanning at least 35 years before and after the change in ICB fire regime, allowing us to calculate reliable statistics.

Both the streamflow analysis and our analysis of drought-related tree mortality are limited by the difficulties in completely controlling for variations in biological and physical characteristics of the basins. Until more basins are exposed to managed wildfire treatments, however, it will be impossible to obtain the data required to explicitly control for the effects of basin characteristics on hydrology and forest health. As such, empirical comparisons between ICB and other basins are necessarily subject to uncertainties due to natural variability.

Overall, our analyses of data from ICB show a variety of lines of evidence pointing to ecohydrological benefits of managed wildfire in this watershed. The most obvious effect is the vegetation changing to be less dominated by forests. The modeled re-organization of soil moisture during summer indicates that these vegetation changes may restructure and influence hydrological processes at the scale of the ICB. It is possible that this restructuring also leads to changes in streamflow yield, although capturing the true magnitude and importance of any such changes requires further study. These changing hydrological processes may also be responsible for the relatively low drought-related mortality seen in ICB and adjacent burned forests during the drought years of 2014 and 2015. It is possible that our observations of soil moisture, streamflow, and forest health could be influenced by other factors besides fire, such as climate and landscape, despite our efforts to account for such factors in our analyses. The fact that these variables are all behaving in ways that are consistent with our hypotheses, however, is suggestive of the influence of the current fire regime.

Scalability

Replicating the resilience of ICB in other basins across the Sierra Nevada is a top land management priority (USDA-FS 2011). Managed wildfire appears to be a promising tool for this purpose. While managed fire is clearly not a suitable tool for all areas, there is potential to expand managed wildfire to meet restoration objectives. The current revisions of the Land and Resource Management Plans for National Forests (NF) in the southern Sierra Nevada propose creating two new ''zones'' for identifying NF lands that emphasize using managed wildfire for resource benefit. Previous plans only allowed for this type of managed wildfire in a few discrete areas; under the proposed revisions, 69–84% of the land in each NF in the southern Sierra Nevada is included in these two zones. This indicates considerable potential for increased use of managed wildfire. An important objective of these programs is to allow fires to burn under a range of fuel moisture and weather conditions, as opposed to the fairly extreme conditions associated with ''escaped'' wildfires that often occur on Forest Service lands due to an emphasis on suppression (Miller and others 2012; North and others 2015). It is worth noting that a majority of the area of the new zones is in watersheds at around 2000 m elevation. This elevation band coincides with a peak in evapotranspiration in the Sierra Nevada, indicating that changes in vegetation water use could be significant for the hydrology of these basins (Goulden and others 2012).

The Sierra Nevada are highly heterogeneous in terms of topography and geology, and thus observations from the ICB should not be directly extrapolated to the entire region. However, 19,100 km^2 of the Sierra Nevada lie in a comparable climate and elevation range to the ICB according to PRISM climate data (Oregon State University 2004) and USGS elevation maps (USGS 2015). Over half of this area is designated wilderness, representing an area in which the risk of wildfires threatening homes or other structures is reduced (University of Montana 2015). Based on this analysis, there is potential to scale up the ICB experiment by nearly 100-fold over some 9800 km^2, or 18% of the total area in the Sierra Nevada.

Conclusion

The Illilouette Creek Basin provides an example of a successful return to a natural fire regime after decades of fire suppression. This transition was achieved without significant negative effects, and has resulted in reduced fire risk, greater resilience to both fire and drought, greater landscape diversity in vegetation and hydrologic terms, and potentially an increase or stabilization of water yields. The resulting landscape is likely closer to the pre-European settlement ecosystems to which Sierra Nevada species are presumably best adapted.

The most preliminary aspect of the results presented here are the hydrological analyses. Despite documenting spatially explicit and vegetation-dependent differences in water availability, the mechanisms driving these differences are yet to be quantified in the ICB, whereas the consequences of changing vegetation composition for runoff yields remain preliminary. Recently installed soil moisture and weather observation stations, along with detailed hydrological modeling, will soon improve our understanding of the changes in ecosystem function induced by managed wildfire, and the implications for the management of Californian forests, water resources, and landscapes.

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